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Modelling of manure production by pigs and NH_3, N_2O and CH_4 emissions. Part II: effect of animal housing, manure storage and treatment practices

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A model has been developed to predict pig manure evolution (mass, dry and organic matter, N, P, K, Cu and Zn contents) and related gaseous emissions (methane (CH_4), nitrous oxide (N_2O) and ammonia (NH_3)) from pig excreta up to manure stored before spreading. This model forms part of a more comprehensive model including the prediction of pig excretion. The model simulates contrasted management systems, including different options for housing (slatted floor or deep litter), outside storage of manure and treatment (anaerobic digestion, biological N removal processes, slurry composting (SC) with straw and solid manure composting). Farmer practices and climatic conditions, which have significant effects on gaseous emissions within each option, have also been identified. The quantification of their effects was based on expert judgement from literature and local experiments, relations from mechanistic models or simple emission factors, depending on existing knowledge. The model helps to identify relative advantages and weaknesses for each system. For example, deep-litter with standard management practices is associated with high-greenhouse gas (GHG) production (+125% compared to slatted floor) and SC on straw is associated with high NH_3 emission (+15% compared to slatted floor). Another important result from model building and first simulations is that farmer practices and the climate induce an intra-system (for a given infrastructure) variability of NH_3 and GHG emissions nearly as high as inter-system variability. For example, in deep-litter housing systems, NH_3 and N_2O emissions from animal housing may vary between 6% and 53%, and between 1% and 19% of total N excreted, respectively. Thus, the model could be useful to identify and quantify improvement margins on farms, more precisely or more easily than current methodologies.

Keywords: manure management, gas emission, environment, modelling, pig

Implications

Manure management is associated with considerable emissions of ammonia and greenhouse gases, which are harmful for the environment. Moreover, a precise knowledge of manure characteristics could improve the economical and environmental impacts of spreading. However, current methodologies are inadequate with respect to proper accounting of local climate and management. This paper presents a model predicting the effects of actual on-farm conditions and climate on manure characteristics and gaseous emissions in the animal housing and during manure storage and treatment. The equations are partially based on expert judgements, and partially on functions parameterised on the basis of literature data.

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Introduction

Manure is an important source of ammonia (NH_3) and greenhouse gases (GHG), with recognised detrimental effects on the environment (Steinfeld *et al.*, 2006). Approximately, 90% of NH_3 emissions would be due to agriculture, in several European countries, 40% of which coming from animal housing and manure storage (Pain *et al.*, 1998; Misselbrook *et al.*, 2000). NH_3 emissions can lead to health issues for farmers and animals. NH_3 emitted may also be transported in the air near the farm or over long distances in ammonium form. This may have detrimental consequences on crops, ecosystem eutrophication or soil acidification (Steinfeld *et al*., 2006). Greenhouse gases emitted from manure are nitrous oxide (N_2O) and methane (CH_4). N_2O is a powerful GHG, which contributes 296-fold more than carbon dioxide (CO_2) to global warming in a 100-year horizon (Intergovernmental Panel on Climate Change (IPCC), 2006). Carbon flows lead to CO_2 and CH_4 emissions, which are other GHG. CH_4 contributes 23-fold more than CO_2 to global warming (IPCC, 2006). In 1995, agriculture in the European union emitted 470 Tg CO_2-equivalents, of which 56% originated from N_2O, 36% from CH_4 and 8% from CO_2 (Freibauer, 2003). Manure in livestock production systems is also a valuable source of nutrient for crops and organic matter (OM) for soils, but its utilisation can also lead to pollutions such as nitrate leaching, P run-off or heavy metal accumulation. In fact, manure management has a key role in the agroecosystem, and should be precisely balanced with crop production. Therefore, the definition of sustainable manure management systems requires a precise evaluation of both harmful gas production and manure characteristics (mass, volume, nutrient contents, OM, etc.) (Burton and Turner, 2003).

Nutrient and matter flows in manure management systems can be evaluated through sampling and analysis, but this approach is expensive and difficult to achieve in commercial conditions. Particularly, gaseous emission measurement is almost impossible in most on-farm situations. Another alternative is the utilisation of mathematical models that can predict these environmental hazards from the available on-farm information. Two main kinds of models can be distinguished: empirically based models use few variables associated with an emission factor (EF), sometimes combined with some additional factors that are applicable for a certain set of conditions. Such models correspond to national or international reviews such as the European Union (EU) emission inventory program (EMEP/CORINAIR) (Dämmgen and Webb, 2006) for NH_3, and IPCC (2006) for GHG. In the IPCC approach (Tier 1, IPCC, 2006), CH_4 and N_2O emissions are estimated with specific EF given per animal category, and country or region. For CH_4, a more detailed approach has been proposed, where EF is related to maximum CH_4 production potential for a given manure type, volatile solid (VS) content of manure, as estimated from animal diet (dry matter (DM) intake, energy digestibility and ash content), and a CH_4 conversion factor that takes climate and type of manure storage into account (Tier 2, IPCC, 2006). However, compared to the wide diversity of practices, empirical models remain imprecise because they do not sufficiently take into account farmer practices and climatic conditions. For example, storage practices are generally grouped according to manure total solids content in solid ($>$20%), semi-solid (10%–20%) and liquid systems ($<$10%), whereas emissions could be contrasted within each category, depending on farmer practices. This could lead to misleading evaluation of some systems (Monteny *et al*., 2001). On the other hand, mechanistic models attempt to describe the processes at detailed levels (molecular or cellular). Emissions are simulated as the result of physicochemical reactions controlled by key factors (such as pH or O_2) and possibly by microbial consumption/production. Such models have been published to predict NH_3 emissions, particularly for slurry in animal housing (Aarnink and Elzing, 1998; Dourmad *et al*., 2008) and/or outside (Berthiaume *et al*., 2005). Mechanistic models have also been developed, to a lesser extent, for GHG. Sommer *et al*. (2004) proposed algorithms to quantify CH_4 emissions from liquid manure during storage, and N_2O emissions from soils after field application of slurry. Models have also been developed for treatment processes, notably biological N removal process (Béline *et al*., 2007), composting process (Sole-Mauri *et al*., 2007) or anaerobic digestion (SAN). However, such mechanistic models are often built from laboratory experiments, and their use remains very limited on a real scale. They often require key parameters that cannot be easily predicted (such as manure pH), which are not available on farm.

Therefore, there is a need to develop intermediary approaches taking into account the effect of farmer practices and climate more precisely than current reference methodologies, and more easily achievable at farm scale than mechanistic models (Nicholson *et al*., 2002; National Research Council (NRC), 2003). Such intermediary models must also be comprehensive and multicriterion, because the introduction of a technology for reducing for instance one source of gaseous pollution may enhance the emission of this gas elsewhere or of another gas (Petersen *et al*., 2007). However, the range of manure management options gets wider, whereas knowledge about their effects remains relatively scattered, because many studies focus on few systems and on one or few particular concerns (e.g. NH_3). Another difficulty for synthesis lies in the numerous different units used in literature (mass of gas per volume (or mass) of manure per time unit, per surface area, per mass of added VS per time unit, as a percentage of corresponding nutrient input, etc.). This limits the comparison and homogenisation of studies.

Objectives

In this context, the purpose of this work was to build a model to simulate manure evolution and gaseous emissions from pig excretion up to late storage just before spreading or export, for contrasted manure management. This model would further support the development at the pig unit level, including a first model predicting animal excretion (Rigolot *et al*., 2010). The objective of this paper is to present the building of the model, with an original approach based on a mix of expert judgement, reference methods, mechanistic models and literature surveys, which could be used in other contexts.

For animal housing, the model must be able to simulate slatted floor with production of slurry or deep litter with production of solid manure (while outdoor production is not included in the present model). Treatment processes included in the model correspond to some of the most widespread and/or promising systems in France and Europe. For slurry, they are biological N removal process and slurry composting (SC) with straw, which represent about 80% and 15% of treatments implemented in French pig farms, respectively (Levasseur *et al.*, 2003). Biological N removal treatment consists in consecutive aerobic and anaerobic processing of slurry allowing N removal by nitrification and denitrification. Slurry composting with straw consists in the spreading of fattening pigs' slurry on straw followed by mixing to produce compost (Guernevez® process). Moreover, the model also includes slurry SAN option, which is becoming increasingly popular in several countries. This process degrades OM of slurry into biogas to produce energy. For solid manure, an additional composting outdoor process is also quite common in pig farms, especially in organic farming.

The model aims at taking into account the main effects of farmer practices for given housing, storage and treatment systems, as well as the effects of climatic conditions. Farmer practices correspond to all possible interventions by the farmer for a given infrastructure. It refers to litter and slurry management inside and outside the building, as well as during solid manure composting (MC).

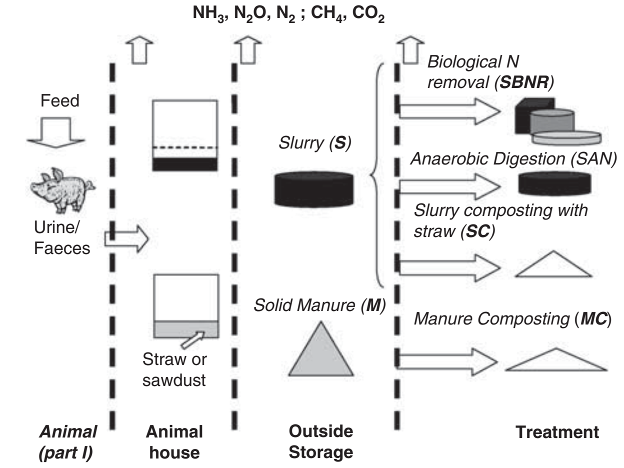

Figure 1 A schematic diagram of the model. Two main options are available for manure management in animal house and outside storage: S, slurry; M, solid manure. Three main options are available for slurry treatment (SBNR, biological nitrogen removal; SAN, anaerobic digestion; SC, slurry composting with straw) and one option for solid: MC, manure composting.

Model description

General description

The model presented in this paper is built in three steps: (i) animal housing (ii) outside storage and (iii) manure treatment. These steps are constructed with a mass-balance approach: nutrient, water and matter amounts in manure are calculated at each step as the difference between inputs and outputs, and the processes at one step depend on what happened at the previous step. Animal housing and outside storage includes two main options: slurry or solid manure management, which are presented separately in this paper. Inputs of the animal housing step are animal excretion (described in Rigolot *et al.*, 2010) and litter. Inputs of the outside storage step are rain and manure transferred from the animal housing. Outputs from both steps are gaseous emissions and manure transferred to the next step. Manure treatment includes four options corresponding to the objectives: biological N removal, SAN, SC with straw and MC (Figure 1). Inputs of manure treatment are manure coming from stores and additional matter and rain, and outputs are gaseous emissions and manure composition before spreading or export. Gaseous emissions are expressed in various units. Particularly, GHG emissions are expressed in kg CO_2eq (carbon dioxide equivalent) by weighting N_2O and CH_4 emissions with their global warming potential for 100 years (IPCC, 2006). Carbon dioxide emissions are not taken into account for this output, considering they are involved in the carbon short-term cycle, with no obvious effect on global warming.

An empiric approach is developed, and the model is built by an expert panel composed of specialists in animal housing and manure storage and treatment. Gaseous emissions are calculated with an EF, weighted by the product of several variation factors (VFs) (equation (1)). Variations factors are identified (qualitatively) and their effect is weighted (quantitatively) by the experts, from published mechanistic models, from the literature or from their own expertise, depending on the available information. A first work during model building was to identify these parameters, and then to quantify them.

$$X_{\text{emitted}}(\text{kg/pig}) = \text{Emission Factor} \times \prod_{i=1}^{n} (\text{Variation Factor}_i). \tag{1}$$

The unit of the EF depends on the nature and/or the knowledge of the emission process. For example, NH_3 EF in the building is expressed as a percentage of excreted N, whereas NH_3 EF in the slurry tank is expressed per square metre, considering it firstly depends on the surface involved. The units of VFs balance the equation, but VFs can also be dimensionless (VF $<$ or $>$ 1 if decreasing or increasing the basic emission, respectively). The effects of VFs are weighted either with continuous equation (e.g. effect of temperature on slurry emissions) or with specific VF value given for different classes (e.g. 'careful', 'normal' or 'careless' litter management). For several VFs, nutrient flow variability is high within a class defined by the experts. For example, the effect of slurry store covering on gaseous emissions depends on cover type. For simplicity and homogeneity reasons, additional classes have not been defined, which define the limitations and applicability of equations. However, the user may fit the model according to the specific situation he has to evaluate.

Emissions and evolution of manure composition in the building and during outside storage

N emissions. N losses from manure may occur mainly as NH_3, N_2O or N_2. Emissions during indoor and outdoor storage are calculated separately, as mentioned in the general description of the model. In the building, the EF corresponds to a volatilisation coefficient applied to N input (i.e. the sum of N excreted and possibly N amount in the added bedding material). For the outside storage period, the EF is expressed in g/m^2 per day for NH_3 when slurry is stored, and in kg N/kg N stored for other gases or when solid manure is stored.

Liquid management of manure. The transcription of equation (1) for NH_3 emission in the buildings with slurry collection is as follows (equation (2)):

$$NH_{3Building}(kg) = 17/14 \times 0.24 \times N_{Excreted} \times VF_{NDilution} \times VF_{Temperature} \times VF_{Air\,Ventilation} \times VF_{Floor} \times VF_{Frequency}. \quad (2)$$

The EF is 0.24 kg N-NH_3/kg N excreted. It was estimated as the difference of measured N in slurry of pigs in 52 experiments and a simulation of raw N excretion (Dourmad *et al*., 1999; Rigolot *et al*., 2010). Factor (17/14) aims at converting the results into kg NH_3. VFs associated with NH_3 volatilisation from the building (Guingand, 1996; Aarnink, 1997) are mainly N excreted (kg), slurry dilution ($VF_{NDilution}$), slurry temperature (VF_{Temp}), air ventilation ($VF_{Ventilation}$), type of floor (VF_{Floor}) and frequency of slurry flushing ($VF_{Frequency}$) (equation (2)). These effects are evaluated through empirical relationships available in the literature or adapted from the model proposed by Aarnink and Elzing (1998).

$VF_{NDilution}$ (equation (2a)): the NH_3 concentration (TAN) of fresh slurry is calculated from urinary N (see part I). In the studies used to estimate the value of EF, average NH_3 concentration of fresh slurry is 0.51 mol/l. For this NH_3 concentration the value of $VF_{NDilution}$ is fixed at 1 and it becomes 0.88 and 1.13 when NH_3 concentration decreases or increases by 20%, respectively (Aarnink and Elzing, 1998).

$$VF_{NDilution} = 1 + 1.27(N_{TAN-Building} - 0.51) \text{ with } N_{TAN-Building} = (N_{Urine}(g)/14(g/mol))/Effluent_{Volume}(l)). \quad (2a)$$

VF_{Temp} (equation (2b)): when slurry temperature is 22°C, which was the average value in the studies reviewed by Dourmad *et al*. (1999), the value of VF_{Temp} is fixed at one. It becomes 0.77 and 1.24 when slurry temperature decreases or increases by 20%, respectively (Aarnink and Elzing, 1998). Slurry temperature is estimated from ambient temperature (T°) using a relationship derived from the study of Granier *et al*. (1996).

$$VF_{Temp} = 1 + 0.053(Temp_{Effluent} - 22) \text{ with } Temp_{Effluent} = -0.012\,T^2 + 1.1813\,T + 1.6064. \quad (2b)$$

$VF_{VentilationRate}$ (equation (2c)): is estimated considering that changes in air speed are proportional to changes of air ventilation rate. When air ventilation rate is of 0.6 m^3/kg BW per hour, which was the average value in the studies reviewed by Dourmad *et al*. (1999), the value of the coefficient $VF_{VentilationRate}$ is fixed at 1. The effect of air ventilation rate on NH_3 volatilisation is then estimated according to Aarnink and Elzing (1998), the coefficient $VF_{VentilationRate}$ becoming 0.92 and 1.08 when air ventilation rates are decreased or increased by 20%, respectively.

$$VF_{VentilationRate} = 1 + 0.636(Rate_{Ventilation} - 0.6). \quad (2c)$$

VF_{Floor} (equation (2d)): a reduction of 20% is retained as default value when the floor is partially slatted, and 15% for metallic slatted floors (Hoeksma *et al*., 1992). However, there is great uncertainty on this value because it was not corroborated by Guingand and Granier (2001).

$$VF_{Floor} = 1.00 \text{ (Concrete Fully Slatted); } 0.85\text{(Metallic Fully Slatted); } 0.80\text{(Partially Slatted)}. \quad (2d)$$

$VF_{Frequency}$ (equation (2e)): Ammonia volatilisation is higher when the slurry is stored for long periods under the animals, as is often the case in France, and decreases when the frequency of slurry removal increases (Hoeksma *et al*., 1992; Voermans and van Poppel, 1993; Guingand, 2000). For instance, a weekly or a daily flushing of the slurry reduces NH_3 emissions by 20% and 35%, respectively, compared to a storage lasting more than 4 weeks.

$$VF_{Frequency} = 1.00(\geqslant 4 \text{ weeks}); 0.90(2 \text{ weeks}); 0.80(1 \text{ week}); 0.65(\leqslant 1 \text{ day}). \quad (2e)$$

Ammonia EF from the outdoor slurry tank is expressed on an area basis (0.6 g NH_3/m^2 per day). This factor and the relationship proposed to integrate VFs is based on Pelletier *et al*. (2006) equation (3).

$$NH_{3Outdoor}(g) = 1.57 \times VF_{Temperature} \times VF_{NDilution} \times VF_{Cover} \times SurfaceArea \times StorageTime. \quad (3)$$

$VF_{Temperature}$ (equation (3a)): is calculated with the following exponential relationship, where slurry temperature is estimated from external air temperature with a relationship found in the same study of Pelletier (Pelletier F, personal Communication).

$$VF_{Temperature} = e^{(0.08 \times TempEffluent)}/2.612 \text{ with } Temp_{Effluent} = 0.9614\,T + 1.6889, \quad (3a)$$

where $R^2 = 0.92$.

$VF_{NDilution}$ (equation (3b)): is a correction factor applied in order to take into account the effect of ammoniacal N concentration (TAN, in mg/kg) on the basis of that (1.03 mg/kg) in the study of Pelletier *et al*. (2006).

$$VF_{NDilution} = TAN(mg/kg)/1.03(mg/kg). \quad (3b)$$

VF_{Cover} is 1 when slurry tank is not covered. For covered tank, 0.2 has been proposed as a default value from the expert's expertise. However, more precise estimation could be used to take into account cover type or wind speed (Olesen and Sommer, 1993). Surface area (m^2) is calculated according to storage time required and height of slurry tank, and storage time is expressed in days.

When slurry is produced, N_2O and N_2 emissions are very low. In the model, N losses in N_2O form are estimated as a fixed percentage (0.2%) of N excreted (in the building), or of N stored (in the slurry tank) (IPCC, 2006). N_2 emission is estimated considering a ratio (N_2O/N_2) equal to 1 : 5 (Loyon *et al*., 2007).

Deep litter system. FEs for total N losses (equation (4)), NH_3 (equation (5)) and N_2O (equation (6)) emissions from the building have been determined from a review of 39 experiments (in Lesguiller *et al*., 1995; Nicks *et al*., 1995; Kaufmann, 1997; Kermarrec, 1999; Robin *et al*., 1999; Texier *et al*., 2000; Texier and Levasseur, 2001; Ramonet and Robin, 2002; Comité d'Orientation pour des Pratiques agricoles respectueuses de l'Environnement (CORPEN), 2003). On average, 64% of N input is emitted into the air, 20% in the form of NH_3 and 6% in the form of N_2O (the remaining in the form of N_2) (Table 1). The effects of the VFs in the building are evaluated from empirical relationships available in the literature or estimated by experts notably in the review used to determine EFs, but also with additional references from De Bode (1991), Groenestein and Van Faassen (1996), Jungbluth *et al*. (2001) and Amon *et al*. (2007). VFs are detailed here for a growing period (30 to 110 kg). They are bedding material type ($VF_{BeddingMaterialType}$), litter surface per animal ($VF_{LitterSurface}$), litter amount ($VF_{LitterAmount}$), litter mixing frequency (VF_{Mixing}) and litter management, which affect litter moisture ($VF_{Management}$). All VFs are dimensionless and their values are given in Table 1.

$$N_{LossesBuilding}(kg) = 0.64 \times N_{Initial} \times VF_{BeddingMaterialType} \times VF_{LitterSurface} \times VF_{Maintenance} \times VF_{Bedding\,materialAmount} \times VF_{Mixing}. \quad (4)$$

$$NH_{3EmittedBuilding}(kg) = 17/14 \times 0.20 \times N_{Initial} \times VF_{LitterSurface} \times VF_{Maintenance} \times VF_{Bedding\,materialAmount}. \quad (5)$$

$$N_2O_{EmittedBuilding}(kg) = 44/28 \times 0.06 \times N_{Initial} \times VF_{BeddingMaterialType} \times VF_{LitterSurface} \times VF_{Maintenance} \times VF_{Bedding\,materialAmount} \times VF_{Mixing}. \quad (6)$$

$VF_{BeddingMaterialType}$: this can be clearly identified from the literature survey. Total N losses correspond to 56% ± 13% and 72% ± 8% of total N, for straw and sawdust based systems, respectively. Bedding material type influences N_2O emissions for both post-weaning and fattening pigs (Nicks *et al*., 2002). In most cases, observed emission was lower for straw and higher for sawdust.

$VF_{LitterSurface}$: when the litter area per animal is large, exchanges with the air increase. This results in higher total N losses. This would be due to higher losses in the form of N_2, because both NH_3 and N_2O emissions are reduced with larger litter area per animal. In the literature survey, NH_3 emissions correspond to between 5% and 15% of total N excreted for large litter area (low-animal density), and between 15% and 25% for usual area. Concerning N_2O, Hassouna *et al*. (2005) mentioned emissions between 2% and 8% for large litter area and between 4% and 12% for usual area.

$VF_{Management}$: a 'careful' management corresponds to particularly appropriate supplies of straw during the growing period (the right amount at the right moment and at the right location in the pen, resulting in a dry litter at the end of the growing period). Careful management is associated with higher total N losses. It also facilitates N immobilisation, which tends to decrease the emissions of NH_3. On the contrary, careless management results in very high emissions of NH_3, up to 60% of initial N in extreme situations (Ramonet and Robin, 2002), because of higher moisture content and inhibition of N immobilisation. N_2O emissions are reduced, because careless management alters nitrification and denitrification processes.

VF_{Mixing}: total N gas emissions increase for a few days after mixing, therefore a VF (+10%) is attributed to frequently mixed litter. Moreover, the effect of frequent mixing on N_2O emissions (due to induced aeration) has been clearly identified (Kermarrec, 1999).

$VF_{BeddingMaterialAmountt}$: a large amount of bedding material (straw or sawdust) may facilitate N immobilisation. However, between supplies of 30 and 100 kg straw per fattening pig (which include most of the practical situations), no effect of the amount of bedding material appears from the literature survey, revealing the predominance of management effect. N_2O emissions are reduced when the

Table 1 *Emission factors and VF for solid manure in pig housing (from literature and expert knowledge)*

	Emissions		Losses
	NH_3-N	N_2O-N	Total N
Emission factor[1] (kg/kg total initial N)	0.20	0.06	0.64
$VF_{BeddingMaterial}$			
Straw	1.0	0.8	0.88
Sawdust	1.0	1.2	1.13
$VF_{LitterSurface}$			
1 m^2/pig	1.1	0.8	1.0
2 m^2/pig	0.5	0.5	1.1
$VF_{Maintenance}$			
Careful (dry litter)	0.8	0.5	1.1
Careless	2.0	0.2	1.0
$VF_{Bedding\ materialAmount}$			
>100 kg/pig	0.8	0.8	0.9
<30 kg/pig	1.2	0.8	1.0
VF_{Mixing}			
Frequent	1	2	1.1

NH_3 = ammonia; N_2O = nitrous oxide; N = nitrogen; VF = variation factor.
[1]Emission factor is the average emission or loss measured among all situations of a 39 experiments review (in CORPEN, 2003).

amount of bedding material is very high (higher N immobilisation) or very low (high-moisture content).

Other factors may influence N_2O emission, such as temperature or microbial composition, but knowledge is still insufficient to propose weighted coefficients.

During outside storage, emissions are estimated as a simple percentage of stored N (no VFs), because experts did not estimate any clear effect of climatic conditions (rain and temperature) either from the literature survey or their own expertise. Assumptions to build the model come from the experiments of Espagnol *et al.* (2006) who found N losses corresponding to 27% of stored N, of which 7.3% in the N-NH_3 form and 3.2% in the N-N_2O form. However, as for slurry tank covering, emission variability is probably high between solid manure storage conditions, and other estimations could be more appropriate for specific conditions (Petersen *et al.*, 1998).

Carbon emissions. CH_4 estimations are based on IPCC Tier 2 methodology (IPCC, 2006, equation (7)), with some adaptations in order to consider the effects of temperature and storage duration inside and outside the building, as proposed by Sommer *et al.* (2004).

$$CH_{4\text{Emitted}}(\text{kg}) = \text{VS} \times \text{B}_0 \times \text{MCF}, \qquad (7)$$

with: VS, VSs, roughly considered as OM amount (kg); B_0, maximum CH_4 producing capacity (m^3/kg DM); methane conversion factor (MCF), CH_4 conversion factor for the management system considered (slurry or deep litter).

VS is an input of the model calculated from excreted OM (given by part I, Rigolot *et al.*, 2010) and B_0 is taken from IPCC (2006). To integrate farmer practices, specific MCF are calculated for slurry management by modulating IPCC default value by temperature and storage time inside and outside the building. The effect of storage time is simply calculated proportionally to reference storage time. Temperature effect is calculated by an Arrhenius relationship parameterised from the study of Vedrenne (2006) (equation (8)).

$$\text{VF}_{\text{Temperature}} = \Phi^{(\text{TempEffluent} - 20)}. \qquad (8)$$

Parameter Φ is a constant estimated as 1.12 by Vedrenne (2006) and $\text{Temp}_{\text{Effluent}}$ is slurry temperature, estimated as previously mentioned in the building and in the outside store.

Manure characteristics after storage. Most manure characteristics are obtained by a mass-balance approach: nutrient and matter amounts. The total weight of the effluent is obtained by adding the amounts of water and DM. However, the prediction of water amount requires additional assumptions. For liquid management, effluent water amount is the sum of excreted water, cleaning water and rainwater, minus water evaporated. Cleaning water is a parameter of the model. Added rainwater and evaporated water are calculated from the surface of the outside store (if not covered) and precipitation amount or potential Penman evaporation, respectively, which are parameters of the model. For solid management, water content is simply calculated from DM amount in the effluent, considering from expert knowledge that DM proportion in solid manure is equal to 40%, 30% and 20% when litter management was 'careful' 'normal' and 'careless', respectively.

Emissions and manure evolution during treatment

For slurry SAN, gaseous emissions and manure characteristics are calculated in two steps: the treatment process itself and the storage of the products. Conversely, for SC, they are calculated in one step, considering that the process continues up to the use of the product. In addition, for biological N removal, the storage of end products is considered as part of the process in this paper, which means that storage time is not a VF, but corresponds to a reference. However, a user of the model could easily integrate own references or additional VFs to parameter the processes. In the special case of MC, emissions are calculated as proposed by equation (1), because they could be easily modified by adopting other composting management practices without any significant infrastructure changes.

Anaerobic digestion of slurry. Manure evolution and gaseous emissions during SAN depend on substrate, retention time and temperature. We consider the case with no gaseous leak during slurry digestion, and therefore N losses are not taken into account. Furthermore, all the carbon in the biogas is considered to be emitted in CO_2 form, when burned to produce energy. The amount of OM converted into biogas during the process can be expressed as a B_0 fraction. Moreover, mineralisation of organic N, which is important during the digestion process, must also be parameterised. For the simulations presented in this paper, SAN parameters were fitted from the study of Vedrenne (2006). In the study, 85% of slurry B_0 was converted into biogas, and 30% of initial organic N turned to NH_3 fraction. During the storage of digested slurry, CH_4 emissions are estimated as 63% lower than those calculated for raw-slurry storage during the same period (Vedrenne, 2006). Other emissions and slurry evolution during post-processing storage are calculated as for raw slurry storage.

Biological N removal. Matter and gaseous flows during aerobic treatment of slurry have been modelled by Béline *et al.* (2004) and Loyon *et al.* (2007), respectively, for the three main treatment schemes found in France. In these schemes, the nitrification and denitrification processes are associated with other slurry manipulations. In a first scheme (scheme 1), raw slurry is directly treated, whereas in the two other schemes (2 and 3), a phase separation is performed ahead, either with a compacting screw (scheme 2), or with a decanter centrifuge (scheme 3). In the three schemes, biological N removal process is followed by a decantation process. Thus, scheme 1 generates two treatment products (biological sludge and supernatant), whereas schemes 2 and 3 generate one more product (separation refusal) because of the phase separation. In this paper, gaseous emissions (Table 2), as well as mass and nutrient flows (Table 3), are given for the

Table 2 *Emission factors associated with the biological N removal process (from Loyon* et al.*, 2007)*

	$N\text{-}NH_3$	$N\text{-}N_2O$	N_2	$C\text{-}CH_4$	$C\text{-}CO_2$
	% of N before treatment[1]			% of C before treatment[1]	
Scheme 1 (no separation)	1.5	0.8	65.9	5.1	9.3
Scheme 2 (compacting screw)	2.3	0.8	62.1	5.2	15.2
Scheme 3 (decanter centrifuge)	3.3	0.7	49.4	5.7	19.6

NH_3 = ammonia; N_2O = nitrous oxide; N_2 = nitrogen; CH_4 = methane; CO_2 = carbon dioxide.
[1]Treatment includes phase separation, biological N removal process, decantation and storage of end products.

Table 3 *Partition of matter and nutrients of slurry between products issued from biological N removal process (as a percentage of the total amounts entering the treatment[1]) (from Béline* et al.*, 2004)*

	Product	Amount	N	P	K	Cu	Zn
Scheme 1 (no separation)	Biological sludge	33.5	22.4	71.1	33.5	87.3	82.5
	Supernatant	66.5	7.6	28.9	66.5	12.7	17.5
Scheme 2 (compacting screw)	Separation refusal	4.75	9.6	53.7	4.8	9.2	10.4
	Biological sludge	31.9	20.3	24.5	31.9	79.3	73.9
	Supernatant	63.3	6.8	21.8	63.3	11.6	15.7
Scheme 3 (decanter centrifuge)	Separation refusal	13.0	33.4	80.6	7.9	31.6	50.2
	Biological sludge	29.1	14.9	13.8	30.9	59.7	41.1
	Supernatant	57.9	5.0	5.6	61.2	8.7	8.7

[1]Treatment includes phase separation, biological N removal process, decantation and storage of end products.

Table 4 *Emission factors associated with slurry composting with straw (from Paillat* et al.*, 2005a and 2005b)*

	$NH_3\text{-}N$	$N_2O\text{-}N$	N_2	$CO_2\text{-}C$	$CH_4\text{-}C$
Emission factor (kg/kg treated)[1]	0.10	0.06	0.44	0.57	0.06

NH_3 = ammonia; N_2O = nitrous oxide; N_2 = nitrogen; CH_4 = methane; CO_2 = carbon dioxide.
[1]Treated corresponds to total N or C input in both treated slurry and the straw required to compost it according to Guernevez process.

whole treatment, including phase separation and storage of end products. The details for each step of the three schemes and the hypotheses corresponding to the given figures are reported in Béline *et al.* (2004).

Liquid composting with straw. Emissions and effluent evolution associated with composting of slurry with straw according to Guernevez® process have been modelled by Paillat *et al.* (2005a and 2005b). The model presented here first calculates the amount of straw required to compost the slurry, on the basis of 12 tonnes slurry per tonne straw. Gaseous emissions and N and matter losses during the whole composting process are calculated as a percentage of treated elements, including both raw slurry and added straw (Table 4). The details for the two main steps of SC (spreading and maturing) are described in Paillat *et al.* (2005b).

Solid manure composting. To apply equation (1) for MC, we have made some hypotheses from Paillat *et al.* (2005b) (15 local experiments) who measured gaseous emissions (CO_2, CH_4, NH_3, N_2O and H_2O) from different mixtures made with various bedding materials in order to cover a wide range of carbon biodegradability, N availability, moisture content and DM density. These are the main factors involved in the composting process (Paillat *et al.*, 2005c; Abd El Kader *et al.*, 2007). These mixtures have been used to define EFs and to calculate the main weighting effects (Table 5): $VF_{ManureType}$, which integrates both C/N ratio and DM content classifications, $VF_{TurningNumber}$, $VF_{OutsideTemperature}$ and $VF_{CompostingDuration}$ (9). As for SC with straw, EFs are given as a percentage of treated elements, including both pig excretion and added straw.

$$X_{EmittedComposting}(kg) = EmissionFactor \times VF_{ManureType} \times VF_{TurningNumber} \times VF_{OutsideTemperature} \times VF_{CompostingDuration}. \quad (9)$$

EFs correspond to the maximum emission measured in experiments with no turning, outside air temperature of about 25°C and a 2-month composting period (Table 5).

$VF_{ManureType}$: manure type refers both to manure moisture and C/N ratio. Three moisture classes as well as three C/N ratio classes were defined. Moisture and C/N ratio classes match, respectively, with the three management classes and the three straw amount classes previously defined in the building.

$VF_{TurningNumber}$: turning has little effect on NH_3 and CH_4 emissions (Espagnol *et al.*, 2006; Abd El Kader *et al.*, 2007)

Table 5 *Emission factors and variation factors for solid manure composting (from Paillat* et al.*, 2005b)*

		Emissions				Losses		
		NH_3-N	N_2O-N	CO_2-C	CH_4-C	total N	H_2O	DM
Emission factor[1]		0.45	0.03	0.45	0.015	0.50	0.75	0.55
$Effect_{ManureType}$								
	DM (%)							
C:N < 15	<25	0.4	1.0	0.6	1.00	1.0	0.8	0.6
	25 to 35	1.0	0.3	0.9	0.03	1.0	0.8	0.9
	>35	0.7	0.1	0.8	0.02	0.8	0.3	0.8
15 < C:N < 25	<25	0.4	0.5	1.0	0.24	1.0	0.7	1.0
	25 to 35	0.8	0.3	1.0	0.04	0.9	1.0	1.0
	>35	0.5	0.1	0.8	0.02	0.5	0.5	0.8
C:N > 25	<25	0.3	0.5	1.0	0.10	0.7	0.6	1.0
	25 to 35	0.5	0.3	1.0	0.02	0.6	0.9	1.0
	>35	0.2	0.1	0.8	0.02	0.2	0.6	0.8
$Effect_{TurningNumber}$								
0		1.0	1.0	1.0	1.0	1.0	1.0	1.0
1		1.0	1.2	1.2	1.0	1.0	1.1	1.2
2		1.0	1.3	1.3	1.0	1.0	1.2	1.3
$Effect_{OutsideTemperature}$								
5		0.7	1.0	0.8	0.8	0.8	0.8	0.8
20		0.9	1.0	1.0	1.0	1.0	1.0	1.0
35		1.2	1.0	1.1	1.1	1.1	1.1	1.1
$Effect_{Composting\ Duration}$								
<2 months		1.0	1.0	1.0	1.0	1.0	1.0	1.0
>6 months		1.1	1.4	1.4	1.4	1.4	1.4	1.4

NH_3 = ammonia; N_2O = nitrous oxide; C = carbon; N = nitrogen; CH_4 = methane; CO_2 = carbon dioxide; DM = dry matter.
[1]Emission factor corresponds to the maximum emission or loss measured among all composting situations for a 2-month period at 25°C without turning.

but significant effect on CO_2 and H_2O emissions and to a lesser extent on N_2O emission (Petersen *et al*., 1998; Espagnol *et al*., 2006; Abd El Kader *et al*., 2007).

$VF_{OutsideTemperature}$: at the beginning of the composting period (thermophilic phase), the influence of outside temperature on gaseous emissions is very low compared with that of heap temperature (Pel *et al*., 1997). When heap temperature decreases under 40°C, the effect of outside temperature could be higher, especially on NH_3 emission (Paillat *et al*., 2005b).

$VF_{CompostingDuration}$: composting period (<2 months or >2 months) has an obvious effect on manure characteristics as it determines the extent of maturation phase, but few studies have dealt with long composting duration. However, we assume that CH_4, CO_2 and H_2O emissions significantly increase with the composting time (Paillat *et al*., 2005b; Pel *et al*., 1997). As regard NH_3, we assume that emissions are less affected by the duration of the composting period, because it mainly occurs during the thermophilic phase (Fukumoto *et al*., 2003).

Simulations

Comparison of structural options for manure management
Scenario definition. Six contrasted manure management systems have been defined as combinations of housing/storage and treatment options (Figure 1): slatted floor and slurry storage combined with no slurry treatment (S), anaerobic digestion (SAN), biological nitrogen removal process (SBNR) or SC with straw and deep litter and solid manure storage combined with no treatment (M) or MC. To compare the six systems on a common basis, six related basic scenarios have been defined, in which farmer practices correspond to most common practices in France, according to expert knowledge. Notably, when the slatted floor option is taken, the slurry is kept for the whole fattening period (100 days) in the building at 22°C, and 30 l of cleaning water are used per pig. When the deep-litter option is taken, 60 kg of straw are used per animal with standard management and animal density (1.2 m^2/pig). Outside storage corresponds to 120 days, average outside temperature is 13°C. The same animal excretion is used as input for the six scenarios. It has been obtained for a growing period from 30 to 110 kg body weight with 'standard' feeding strategy and performance parameters, as defined in the companion paper (raw excretion per pig: volume = 384 l, N = 3.82 kg, P = 5.8 kg, K = 16.2 kg, Cu = 5.6 g and Zn = 32.6 g Rigolot *et al*., 2010).

Simulation results. Ammonia, N_2O and CH_4 emissions in kg/pig, as well as total GHG emissions, expressed in kg CO_2eq (carbon dioxide equivalent), are presented in Table 6. With the chosen assumptions for farmer practices in the basic scenarios, housing the animals on straw reduces NH_3 emissions compared to slatted floor, when no additional treatment is achieved (−11%, S *v.* M). Systems including a composting process increase NH_3 emissions (+15% SC *v.* S, and +45% MC *v.* M). Conversely, SAN and biological N removal treatment both reduce NH_3 emissions compared to classic slurry storage (−4% and −8%, respectively).

Total GHG emissions are very contrasted between slurry-based scenarios (S, SAN and SBNR) and straw-based scenarios (SC, M and MC); the last possibly inducing emissions between two and three-fold higher. The contribution of N_2O and CH_4 to this difference is antagonist, as mentioned by Monteny *et al.*, (2001): CH_4 emission is higher in anaerobic systems (S) and N_2O is mainly produced in aerated systems (M and MC), whereas in systems where both conditions may appear (SC), large amounts of both gases may be emitted. However, because of the much higher global warming potential of N_2O, total GHG emissions are much higher in aerobic (litter-based) systems.

Product characteristics (mass, nutrients and trace element contents and DM and OM contents) are given in Table 7 as a quantitative illustration of manure evolution. As regard nutrient content, total N amount in products vary considerably between systems, whereas P and K total amounts only depend on straw addition, because they are completely kept in manure, whatever the system. N amounts make it possible to calculate N abatement in gaseous form (as a percentage of excreted N) for each system: S (28.6%), SAN (28.0%), SBNR (72.3%), SC (67.7%), M (72.6%) and MC (79.0%).

Table 6 *Total gaseous emissions simulated by the model for six scenarios of manure management, for one fattening pig (30 to 110 kg BW)*

	NH_3	N_2O kg/pig[1]	CH_4	GHG kg CO_2eq/pig
S	1.29	0.01	2.78	67.1
SAN	1.24	0.01	1.93	47.5
SBNR	1.19	0.04	2.40	67.9
SC	1.48	0.29	4.27	185.2
M	1.14	0.49	0.21	150.8
MC	1.65	0.44	0.24	136.1

NH_3 = ammonia; N_2O = nitrous oxide; CH_4 = methane; S = slurry; SAN = slurry + anaerobic digestion; SBNR = slurry + biological nitrogen removal; SC = slurry composting; M = solid manure; MC = solid manure + composting; GHG = greenhouse gas; CO2eq = carbon dioxide equivalent.
[1]from 30 to 110 kg BW.

Effects of feeding strategy and farmer practices on emissions from building and stores

A simple example is given to illustrate how the model might be used to assess the impact of farmer practices. In this example, advised manure management scenarios have been defined by the experts for systems S and M. For system S, advised scenario consists in daily slurry flushing from the building, and the use of partially slatted floor. For system M, advised scenario consists in good litter management and low-animal density. Then, both standard and advised manure management scenarios have been combined with two sets of excretion characteristics, the first corresponding to the 'standard' feeding strategy previously defined (3.82 kg N excreted/growing pig), and the second to an 'environment friendly' feeding strategy, also reported in the companion paper (2.90 kg N excreted/growing pig, Rigolot *et al.*, 2010). Results for NH_3 and GHG emissions are presented in Table 8.

In system M, the use of the advised instead of the standard litter management results in a reduction of about 54% and 62% of NH_3 and GHG productions, respectively. In the same way, in system S, the daily flushing of slurry, compared to storage below the slatted floor, results in 26% and 21% reduction of NH_3 and GHG emissions, respectively. When both advised feeding and advised manure management are practiced, large decreases of NH_3 emissions would be expected (40% and 63% in S and M systems, respectively), as well as large decreases of GHG emissions (36% and 70% in S and M systems, respectively).

Numerous sensitivity analyses and other scenario comparisons, not shown in this paper, have been performed. They confirm that the emissions within each system highly depend on farmer practices. For example, when testing every combination of VF in litter-based housing systems, NH_3 emissions in animal housing may vary between 6% and 53% of total N excreted and that of N_2O between 1% and 19%.

Table 7 *Product characteristics simulated by the model for six scenarios of manure management, for one fattening pig (30 to 110 kg BW)*

	Product	Mass kg/pig[1]	DM	OM	N g/kg	P	K	Cu	Zn mg/kg
S	Slurry	407	76	50	6.7	1.4	4.0	14	80
SAN	Digested slurry	393	44	23	7.0	1.4	4.1	14	83
SBNR	Separation refusal	19	306	265	14.0	16.0	4.0	28	182
	Biological sludge	131	34	14	4.5	1.1	3.9	37	192
	Supernatant	260	7	1	0.8	0.5	3.9	3	21
SC	Compost	147	339	125	8.4	4.4	13.9	39	224
M	Solid manure	291	299	244	3.6	2.4	8.2	19	112
MC	Compost	89	435	352	9.0	7.8	26.7	64	372

DM = dry matter; OM = organic matter; S = slurry; SAN = slurry + anaerobic digestion; SBNR = slurry + biological nitrogen removal; SC = slurry composting; M = solid manure; MC = solid manure + composting.
[1]from 30 to 110 kg BW.

Table 8 *Gaseous emissions in system S and M with different feeding and manure management practices*

Manure management system	Feeding strategy	Manure management practices	Gaseous emissions	
			NH_3 kg/pig[1]	GHG emissions $kgCO_2eq$/pig
S slatted floor + slurry storage	Standard	Flushing after 100 days	1.29	67.0
		Daily flushing	1.02	42.7
	Environment friendly	Flushing after 100 days	0.98	67.0
		Daily flushing	0.77	42.7
M deep litter + solid manure storage	Standard	Standard management + 1.2 m^2/pig	1.14	150.8
		Careful management + 2 m^2/pig	0.53	57.4
	Environment friendly	Standard management + 1.2 m^2/pig	0.89	118.6
		Careful management + 2m^2/pig	0.42	45.8

NH_3 = ammonia; GHG = greenhouse gas; CO2eq = carbon dioxide equivalent; S = slurry; M = solid manure.
[1]from 30 to 110 kg BW.

Validation
A comparison between some simulation results and external data (not used to build the model) has been performed as far as possible. However, a real validation is difficult to perform, because information is often missing in protocol descriptions. In fact, all the inputs required to run the model are generally not given in the material and method sections of publications (e.g. feed characteristics). Therefore, a validation step has been performed by expert knowledge and by carefully checking simulation coherency. For example, estimated total N losses are always higher than or equal to the sum of calculated N-NH_3 and N-N_2O emissions. Furthermore, indicators such as DM contents or C/N ratio in the products, or matter conservation have also been used to verify the consistency of the model.

A limit to the approach proposed (equation (1)) is that the VFs identified by the experts are not necessarily independent. For example, in equation (2) (NH_3 emissions from slurry), an increase in temperature will result in an increase in air renewal and water evaporation, and therefore in NH_3–N concentration. This means that the end user of the model must ensure that the input data describing the system are consistent with each other. In fact, the approach developed in this paper could help to design some multifactorial experiments, or to calibrate mechanistic models, which would be required for a more satisfactory validation of the model.

Discussion and conclusion

This model predicts gaseous emissions and nutrient flows in contrasted manure management systems. The objective was to take into account the main effects of farmer practices and climatic conditions in each system, more precisely than in current methodologies. The construction of the model, as well as its limits, reveals some gaps and improvement margins in the knowledge of such systems. For example, the model points out a general lack of experimental data, but also specific needs for data acquisition, especially for alternative housing systems. Emission measurement reports and databases could also be further improved. Particularly, scientific papers should be as complete as possible for VFs identified in this work, notably concerning animals, manure storage dimensions and duration, and farmer practices. In fact, the experimental support for the different equations presented in this paper is highly variable. Because of trade off between improved details and loss of transparency, it would be useful to quantify uncertainties of estimates for each source, and for emission estimates at the system level.

Modelling helps us to identify environmental advantages and weaknesses concerning NH_3 and GHG emissions and manure characteristics. As regard gaseous emissions, the simulation results (Table 6) highlight some risks associated with some systems for particularly high NH_3 (composting process) and/ or GHG production (litter-based systems). Synergies (reduction of both NH_3 and N_2O) and trade-off (compromise between N_2O and CH_4 emissions) can also be identified.

Another important conclusion of this work is that variations in NH_3 and GHG emissions within systems (for a given infrastructure) might be as high as variations between the main systems, as illustrated by Table 8. For example, in straw-based systems, the combination of VFs (N_2O emission between 1% and 19% of N input) indicates that these systems should not be systematically associated with high-GHG emissions, in spite of the high level measured in standard conditions. They should rather be considered as risky systems, because they might induce lower emissions (whether well managed). Moreover, in these systems, improvements could be achieved without structure modification, provided farmers were sensitised for litter management, and litter management better understood.

At a larger scale, the nature of products and their composition determine opportunities and constraints to their use. In particular, DM content of products will determine possibilities and flexibility of use, on the farm or elsewhere (Burton and Turner, 2003). In an extreme case, composting process could be a total treatment whether the compost is exported. However, export of by-products or their agronomic use or environmental impacts will also depend on their nutrient and trace element contents. Particularly, dry products are generally associated to high levels of trace elements and P, which may limit their use (Dourmad and Jondreville, 2008). Discrepancies between relative proportions of N, P and K in manure and crop average requirements (N/P/K $\approx$ 100/40/70) may lead to high-K application rates and to P accumulation in soils. Moreover, the proportion of organic and ammoniacal N influences NH_3 volatilisation (Petersen *et al*., 2007). For example, NH_3 emission during spreading of digested slurry could be 15% higher compared to raw slurry, because of increased ammoniacal N content (Vedrenne, 2006), whereas compost spreading induces low emissions. This raises the problem of the definition of system boundaries for environmental assessment. More generally, agricultural practices, soil status and climate, the manure market and available equipment for liquid or solid manure management in the farming system, as well as other considerations like animal welfare, will ultimately determine the most suitable manure management system. This will be studied by integrating the results of this study in a comprehensive model at farm scale, including the whole pig production unit and dairy and crop productions (Chardon *et al*., 2007).

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